5. Agriculture

Agricultural activities contribute directly to emissions of greenhouse gases through a variety of processes. This chapter provides an assessment of non-carbon dioxide emissions from the following source categories: enteric fermentation in domestic livestock, livestock manure management, rice cultivation, agricultural soil management, and agricultural residue burning (see Figure 5-1). Carbon dioxide emissions and removals from agriculture-related land-use activities, such as conversion of grassland to cultivated land, are discussed in the Land-Use Change and Forestry chapter.

In 1999, agricultural activities were responsible for emissions of 488.8 Tg CO_2 Eq., or 7.2 percent of total U.S. greenhouse gas emissions. Methane (CH_4) and nitrous oxide (N_2O) were the primary greenhouse gases emitted by agricultural activities. Methane emissions from enteric fermentation and manure management represent about 21 and 6 percent of total CH_4 emissions from anthropogenic activities, respectively. Of all domestic animal types, beef and dairy cattle were by far the largest emitters of methane. Rice cultivation and agricultural crop residue burning were minor sources of methane. Agricultural soil management activities such as fertilizer application and other cropping practices were the largest source of U.S. N_2O emissions, accounting for 69 percent. Manure management and agricultural residue burning were also smaller sources of N_2O emissions.

Figure 5-1

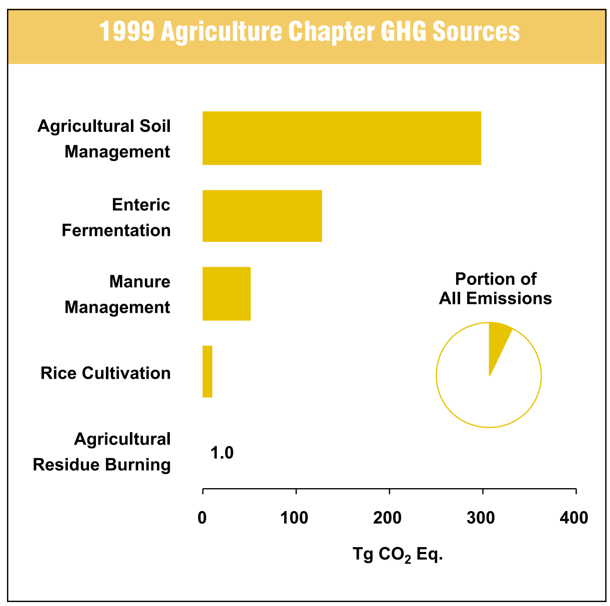


Table 5-1 and Table 5-2 present emission estimates for the Agriculture chapter. Between 1990 and 1999, CH_4 emissions from agricultural activities increased by 4.7 percent while N_2O emissions increased by 10.7 percent. In addition to CH_4 and N_2O, agricultural residue burning was also a minor source of the criteria pollutants carbon monoxide (CO) and nitrogen oxides (NO_x).

Table 5-1: Emissions from Agriculture (Tg CO_2 Eq.)

Gas/Source	1990		1995	1996	1997	1998	1999
CH_4	**165.1**		**177.4**	**172.3**	**172.4**	**173.4**	**172.9**
Enteric Fermentation	129.5		136.3	132.2	129.6	127.5	127.2
Manure Management	26.4		31.0	30.7	32.6	35.2	34.4
Rice Cultivation	8.7		9.5	8.8	9.6	10.1	10.7
Agricultural Residue Burning	0.5		0.5	0.6	0.6	0.6	0.6
N_2O	**285.4**		**302.1**	**311.8**	**317.4**	**317.9**	**315.9**
Agricultural Soil Management	269.0		285.4	294.6	299.8	300.3	298.3
Manure Management	16.0		16.4	16.8	17.1	17.2	17.2
Agricultural Residue Burning	0.4		0.4	0.4	0.4	0.5	0.4
Total	**450.5**		**479.5**	**484.1**	**489.8**	**491.4**	**488.8**

Note: Totals may not sum due to independent rounding.

Table 5-2: Emissions from Agriculture (Gg)

Gas/Source	1990		1995	1996	1997	1998	1999
CH_4	**7,862**		**8,446**	**8,205**	**8,208**	**8,259**	**8,232**
Enteric Fermentation	6,166		6,492	6,295	6,172	6,072	6,057
Manure Management	1,256		1,477	1,463	1,553	1,677	1,638
Rice Cultivation	414		452	419	455	481	509
Agricultural Residue Burning	25		24	28	29	30	28
N_2O	**921**		**975**	**1,006**	**1,024**	**1,026**	**1,019**
Manure Management	52		53	54	55	55	55
Agricultural Soil Management	868		921	950	967	969	962
Agricultural Residue Burning	1		1	1	1	1	1

Note: Totals may not sum due to independent rounding.

Enteric Fermentation

Methane (CH_4) is produced as part of normal digestive processes in animals. During digestion, microbes resident in an animal's digestive system ferment food consumed by the animal. This microbial fermentation process, referred to as enteric fermentation, produces methane as a by-product, which can be exhaled or eructated by the animal. The amount of methane produced and excreted by an individual animal depends primarily upon the animal's digestive system, and the amount and type of feed it consumes.

Among domesticated animal types, ruminant animals (e.g., cattle, buffalo, sheep, goats, and camels) are the major emitters of methane because of their unique digestive system. Ruminants possess a rumen, or large "fore-stomach," in which microbial fermentation breaks down the feed they consume into products that can be utilized by the animal. The microbial fermentation that occurs in the rumen enables them to digest coarse plant material that non-ruminant animals cannot. Ruminant animals, consequently, have the highest methane emissions among all animal types.

Non-ruminant domesticated animals (e.g., pigs, horses, mules, rabbits, and guinea pigs) also produce methane emissions through enteric fermentation, although this microbial fermentation occurs in the large intestine. These non-ruminants have significantly lower methane emissions on a per-animal basis than ruminants because the capacity of the large intestine to produce methane is lower.

In addition to the type of digestive system, an animal's feed intake also affects methane emissions. In general, a higher feed intake leads to higher methane emissions. Feed intake is positively related to animal size, growth rate, and production (e.g., milk production, wool growth, pregnancy, or work). Therefore, feed intake varies among animal types as well as among different management practices for individual animal types.

Methane emission estimates from enteric fermentation are shown in Table 5-3 and Table 5-4. Total livestock methane emissions in 1999 were 127.2 Tg CO_2 Eq. (6,057 Gg) decreasing slightly since 1998. Beef cattle remain the largest contributor of methane emissions from enteric fermentation, accounting for 75 percent of emissions in 1999. Emissions from dairy cattle in 1999 accounted for 21 percent of total emissions, and the remaining 4 percent of emissions can be attributed to horses, sheep, swine, and goats.

Methodology

Livestock emission estimates fall into two categories: cattle and other domesticated animals. Cattle, due to their large population, large size, and particular digestive characteristics, account for the majority of methane emissions from livestock in the United States. Cattle production systems in the United States are better characterized in comparison with other livestock management systems. A more detailed methodology (i.e., IPCC Tier 2) was therefore applied to estimating emissions for cattle. Emission estimates for other domesticated animals were handled using a less detailed approach (i.e., IPCC Tier 1).

While the large diversity of animal management practices cannot be precisely characterized and evaluated, significant scientific literature exists that describes the quantity of methane produced by individual ruminant animals, particularly cattle. A detailed model that incorporates this information and other analyses of livestock population, feeding practices and production characteristics was used to estimate emissions from cattle populations.

The methodology for estimating emissions from enteric fermentation involves the four steps indicated below.

Step 1: Characterize the Cattle Population

National cattle population statistics were disaggregated into the following cattle sub-populations:

Dairy Cattle

- Calves
- Heifer Replacements
- Cows

Table 5-3: CH_4 Emissions from Enteric Fermentation (Tg CO_2 Eq.)

Livestock Type	1990	1995	1996	1997	1998	1999
Beef Cattle	94.7	103.0	100.4	97.8	95.8	95.4
Dairy Cattle	28.7	27.5	26.1	26.0	25.9	26.1
Horses	2.1	2.3	2.3	2.3	2.3	2.3
Sheep	1.9	1.5	1.4	1.3	1.3	1.2
Swine	1.7	1.9	1.8	1.8	2.0	1.9
Goats	0.3	0.3	0.3	0.2	0.2	0.2
Total	**129.5**	**136.3**	**132.2**	**129.6**	**127.5**	**127.2**

Note: Totals may not sum due to independent rounding.

Table 5-4: CH_4 Emissions from Enteric Fermentation (Gg)

Livestock Type	1990	1995	1996	1997	1998	1999
Beef Cattle	4,511	4,902	4,781	4,658	4,561	4,544
Dairy Cattle	1,369	1,308	1,241	1,240	1,234	1,245
Horses	102	108	109	111	111	111
Sheep	91	72	68	64	63	58
Swine	81	88	84	88	93	89
Goats	13	12	13	11	10	10
Total	**6,166**	**6,492**	**6,295**	**6,172**	**6,072**	**6,057**

Note: Totals may not sum due to independent rounding.

Beef Cattle

- Calves
- Heifer Replacements
- Heifer and Steer Stockers
- Animals in Feedlots
- Cows
- Bulls

Calf birth estimates, end of year population statistics, detailed feedlot placement information, and slaughter weight data were used in the model to initiate and track cohorts of individual animal types having distinct emissions profiles. The key variables tracked for each of the cattle population categories are described in Annex J. These variables include performance factors such as pregnancy and lactation as well as average weights and weight gain.

Step 2: Characterize Cattle Nutrition

Diet characteristics were estimated by State and region for U.S. dairy, beef, and feedlot cattle, and were used to calculate Digestible Energy (DE) values and methane conversion rates (Y_m) for each population category. The IPCC recommends Y_m values of 3.5 to 4.5 percent for feedlot cattle and 5.5 to 6.5 percent for all other cattle. Given the availability of detailed diet information for different regions and animal types in the United States, DE and Y_m values unique to the United States were developed, rather than using the recommended IPCC values. The diet characterizations and estimation of DE and Y_m values were based on contact with State agricultural extension specialists, a review of published forage quality studies, expert opinion, and modeling of animal physiology. See Annex J for more details on the method used to characterize cattle diets in the United States.

Step 3: Determine Cattle Emissions

In order to estimate methane emissions from cattle, the population was divided into region, age, sub-type (e.g., calves, heifer replacements, cows, etc.), and production (i.e., pregnant, lactating, etc.) groupings to more fully capture any differences in methane emissions from these animal types. Cattle diet characteristics developed under Step 2 were used to develop regional emission factors for each sub-category. Tier 2 equations from IPCC (2000) were used to produce methane emission factors for the following cattle types: dairy cows, beef cows, dairy replacements, beef replacements, steer stockers, heifer stockers, steer feedlot animals, heifer feedlot animals, and steer and heifer feedlot step-up diet animals. To estimate emissions from cattle, population data were multiplied by the emission factor for each cattle type. More details can be found in Annex J.

Step 4: Determine Other Livestock Emissions

Emission estimates for other animal types were based upon average emission factors representative of entire populations of each animal type. Methane emissions from these animals accounted for a minor portion of total methane emissions from livestock in the United States from 1990 through 1999. Also, the variability in emission factors for each of these other animal types (e.g. variability by age, production system, and feeding practice within each animal type) is less than that for cattle.

See Annex J for more detailed information on the methodology and data used to calculate methane emissions from enteric fermentation.

Data Sources

Annual cattle population data were obtained from the U.S. Department of Agriculture's National Agricultural Statistics Service (1995a-d, 1996b, 1997, 1998a, 1999a-c,f-g, 2000a,c,d). DE and Y_m values were used to calculate emissions from cattle populations. DE and Y_m for dairy and beef cows, and for beef stockers, were calculated from diet characteristics using a model simulating ruminant digestion in growing and/or lactating cattle (Donovan and Baldwin 1999). For feedlot animals, DE and Y_m values recommended by Johnson (1999) were used. Values from EPA (1993) were used for dairy replacement heifers. Weight data were estimated from Feedstuffs (1998), Western Dairyman (1998), and expert opinion. Annual livestock population data for other livestock types, except horses, as well as feedlot placement information were obtained from the U.S. Department of Agriculture's National Agricultural Statistics Service (USDA 1994a-b, 1998b-c, 1999d,e,h, 2000b,e). Horse data were obtained from the Food and Agriculture Organization (FAO) statistical database (FAO 2000). Methane emissions from sheep, goats, pigs, and horses were estimated by using emission factors utilized in Crutzen et al. (1986). These emission factors are representative of typical animal sizes, feed intakes, and feed characteristics in developed countries. The methodology is the same as that recommended by IPCC (IPCC/UNEP/OECD/IEA 1997, IPCC 2000).

Uncertainty

The basic uncertainties associated with estimating emissions from enteric fermentation are the range of emission factors possible for the different animal types and the number of animals with a particular emissions profile that exist during the year. Although determining an emission factor for all possible cattle sub-groupings and diet characterizations in the United States is not possible, the enteric fermentation model that was used estimates the likely emission factors for the major animal types and diets. The model generates estimates for dairy and beef cows, dairy and beef replacements, beef stockers, and feedlot animals. The analysis departs from the recommended IPCC (2000) DE and Y_m values to account for diets for these different animal types regionally. Based on expert opinion and peer reviewer recommendations, it is believed that the values supporting the development of emission factors for the animal types studied are appropriate for the situation in the United States.

In addition to the uncertainty associated with developing emission factors for different cattle population categories based on estimated energy requirements and diet characterizations, there is uncertainty in the estimation of animal populations by animal type. The model estimates the movement of animal cohorts through the various monthly age and weight classes by animal type. Several inputs affect the precision of this approach, including estimates of births by month, weight gain of animals by age class, and placement of animals into feedlots based on placement statistics and slaughter weight data. However, it is believed that the model accurately characterizes the U.S. cattle population and fully captures the potential differences in emission factors between different animal types.

Manure Management

The management of livestock manure can produce anthropogenic methane (CH_4) and nitrous oxide (N_2O) emissions. Methane is produced by the anaerobic decomposition of manure. Nitrous oxide is produced as part of the nitrogen cycle through the nitrification and denitrification of the organic nitrogen in livestock manure and urine.

When livestock or poultry manure are stored or treated in systems that promote anaerobic conditions (e.g., as a liquid in lagoons, ponds, tanks, or pits), the decomposition of materials in the manure tends to produce CH_4. When manure is handled as a solid (e.g., in stacks or pits) or deposited on pasture, range, or paddock lands, it tends to decompose aerobically and produce little or no CH_4. A number of other factors related to how the manure is handled also affect the amount of CH_4 produced: 1) ambient temperature and moisture affect the amount of CH_4 produced because they influence the growth of the bacteria responsible for methane formation; 2) methane production generally increases with rising temperature and residency time; and 3) for non-liquid based manure systems, moist conditions (which are a function of rainfall and humidity) favor CH_4 production. Although the majority of manure is handled as a solid, producing little CH_4, the general trend in manure management, particularly for large dairy and swine producers, is one of increasing use of liquid systems. In addition, use of daily spread systems at smaller dairies is decreasing, due to new regulations limiting the application of manure nutrients, which has resulted in an increase of manure managed and stored on site at these smaller dairies.

The composition of manure also affects the amount of methane produced. Manure composition varies by animal type and diet. The greater the energy content and digestibility of the feed, the greater the potential for CH_4 emissions. For example, feedlot cattle fed a high energy grain diet generate manure with a high CH_4-producing capacity. Range cattle fed a low energy diet of forage material produce manure with about 70 percent of the CH_4-producing potential of feedlot cattle manure. In addition, there is a trend in the dairy industry for dairy cows to produce more milk per year. These high-production milk cows tend to produce more volatile solids in their manure as milk production increases, which increases the probability of CH_4 production.

The production of nitrous oxide from livestock manure depends on the composition of the manure and urine, the type of bacteria involved in the process, and the amount of oxygen and liquid in the manure system. For N_2O emissions to occur, the manure must first be handled aerobically where ammonia nitrogen is converted to nitrites (nitrification), and then handled anaerobically where the nitrite is converted to N_2O (denitrification). These emissions are most likely to occur in dry manure handling systems that have aerobic conditions, but can also undergo saturation to create pockets of anaerobic conditions. For example, manure at cattle drylots is deposited on soil, oxidized to nitrite and nitrate nitrogen, and has the potential to encounter saturated conditions following rain events.

Certain N_2O emissions are accounted for and discussed under Agricultural Soil Management. These are emissions from livestock manure and urine deposited on pasture, range, or paddock lands, as well as emissions from manure and urine that is spread onto fields either directly as "daily spread" or after it is removed from manure management systems (e.g., lagoon, pit, etc.)

Table 5-5, Table 5-6, and Table 5-7 provide estimates of CH_4 and N_2O emissions from manure management by animal category. Estimates for methane emissions in 1999 were 34.4 Tg CO_2 Eq. (1,638 Gg), 30 percent above emissions in 1990. The majority of the increase in methane emissions over the time series was from swine and dairy cow manure and is attributed to shifts by the swine and dairy industries towards larger facilities. Larger swine and dairy farms tend to use flush or scrape liquid systems to manage and store manure. Thus the shift towards larger facilities is translated into an increasing use of liquid manure management systems. This shift was accounted for by incorporating State-specific weighted methane conversion factor (MCF) values calculated from the 1992 and 1997 farm-size distribution reported in the *Census of Agriculture* (USDA 1999e). In 1999, swine CH_4 emissions decreased from 1998 due to a decrease in swine animal populations.

As stated previously, dairies are moving away from daily spread systems. Therefore, more manure is managed and stored on site, contributing to additional CH_4 emissions over the time series. The CH_4 estimates also account for changes in volatile solids production from dairy cows correlated to their generally increasing milk production. A description of the methodology is provided in Annex K.

Total N_2O emissions from manure management systems in 1999 were estimated to be 17.2 Tg CO_2 Eq. (55 Gg). The 7 percent increase in N_2O emissions from 1990 to 1999 can be partially attributed to a shift in the poultry industry away from the use of liquid manure management systems, in favor of litter-based systems and high rise houses. In addition, there was an overall increase in the population of poultry and swine, although swine populations declined slightly in 1993, 1995, 1996, and 1999 from previous years. The population of beef cattle in feedlots, which tend to store and manage manure on site, also increased.[1] Although dairy cow populations went down overall, the population of dairies managing and storing manure on site—as opposed to using pasture, range, or paddock or daily spread systems—went up. Therefore, the increase in dairies using on-site storage to manage their manure results in increased N_2O emissions. As stated previously, N_2O emissions from livestock manure deposited on pasture, range, or paddock land and manure immediately applied to land in daily spread systems are accounted for under Agricultural Soil Management.

Methodology

The methodologies presented in *Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories* (IPCC 2000) form the basis of the CH_4 and N_2O emissions estimates for each animal type. The calculation of emissions requires the following information:

[1] Methane emissions were mostly unaffected by this increase in the beef cattle population because feedlot cattle primarily use solid storage systems, which produce little methane.

Table 5-5: CH_4 and N_2O Emissions from Manure Management (Tg CO_2 Eq.)

Animal Type	1990	1995	1996	1997	1998	1999
CH_4	**26.4**	**31.0**	**30.7**	**32.6**	**35.2**	**34.4**
Dairy Cattle	8.9	11.1	11.2	11.8	12.2	12.5
Beef Cattle	3.2	3.5	3.4	3.4	3.4	3.3
Swine	11.1	13.2	12.8	14.1	16.2	15.3
Sheep	0.1	0.1	+	+	+	+
Goats	+	+	+	+	+	+
Poultry	2.6	2.6	2.6	2.6	2.7	2.6
Horses	0.6	0.6	0.6	0.7	0.7	0.7
N_2O	**16.0**	**16.4**	**16.8**	**17.1**	**17.2**	**17.2**
Dairy Cattle	4.2	4.0	3.9	3.9	3.8	3.8
Beef Cattle	4.9	5.3	5.1	5.4	5.5	5.5
Swine	0.3	0.3	0.3	0.4	0.4	0.4
Sheep	+	+	+	+	+	+
Goats	+	+	+	+	+	+
Poultry	6.3	6.5	7.2	7.2	7.2	7.2
Horses	0.2	0.2	0.2	0.2	0.2	0.2
Total	**42.4**	**47.4**	**47.5**	**49.7**	**52.4**	**51.6**

+ Does not exceed 0.05 Tg CO_2 Eq.
Note: Totals may not sum due to independent rounding.

Table 5-6: CH_4 Emissions from Manure Management (Gg)

Animal Type	1990	1995	1996	1997	1998	1999
Dairy Cattle	422	527	532	561	583	593
Beef Cattle	150	165	164	162	160	159
Swine	527	630	610	670	770	728
Sheep	3	2	2	2	2	2
Goats	1	1	1	1	1	1
Poultry	125	122	123	126	130	124
Horses	29	31	31	31	31	31
Total	**1,256**	**1,477**	**1,463**	**1,553**	**1,677**	**1,638**

Note: Totals may not sum due to independent rounding.

Table 5-7: N_2O Emissions from Manure Management (Gg)

Animal Type	1990	1995	1996	1997	1998	1999
Dairy Cattle	14	13	13	12	12	12
Beef Cattle	16	17	16	17	18	18
Swine	1	1	1	1	1	1
Sheep	+	+	+	+	+	+
Goats	+	+	+	+	+	+
Poultry	20	21	23	23	23	23
Horses	1	1	1	1	1	1
Total	**52**	**53**	**54**	**55**	**55**	**55**

+ Does not exceed 0.5 Gg
Note: Totals may not sum due to independent rounding.

- Animal population data (by animal type and State)
- Amount of nitrogen produced (amount per head times number of head)
- Amount of volatile solids produced (amount per head times number of head)
- Methane producing potential of the volatile solids (by animal type)
- Extent to which the methane producing potential is realized for each type of manure management system (by State and manure management system)
- Portion of manure managed in each manure management system (by State and animal type)
- Portion of manure deposited on pasture, range, or paddock or used in daily spread systems

Both CH_4 and N_2O emissions were estimated by first determining activity data, including animal population, waste characteristics, and manure management system usage. For swine and dairy cattle, manure management system usage was determined for different farm size categories using data from USDA (USDA 1996b, 1998d, 2000h) and EPA (ERG 2000). For beef cattle and poultry, manure management system usage data was not tied to farm size (ERG 2000, USDA 2000i). For other animal types, manure management system usage was based on previous EPA estimates (EPA 1992).

Next, "base" methane conversion factors (MCFs) and N_2O emissions factors were determined for all manure management systems. Base MCFs for dry systems and base N_2O emission factors for all systems were set equal to default IPCC factors (IPCC 2000). Base MCFs for liquid/slurry and deep pit systems were calculated using the average annual ambient temperature for the climate zone where the animal populations are located. For anaerobic lagoon systems, the base MCFs were calculated based on the average monthly ambient temperature, the carryover of volatile solids in the system from month to month due to long storage times exhibited by these systems, and a factor to account for management and design practices that result in the loss of volatile solids from the system.

For each animal group—except sheep, goats, and horses—the base emission factors were weighted to incorporate the distribution of management systems used within each State to create an overall State-specific weighted emission factor. To calculate this weighted factor, the percent of manure for each animal group managed in a particular system in a State was multiplied by the emission factor for that system and State, and then summed for all manure management systems in the State.

Methane emissions were estimated by calculating the volatile solids (VS) production for all livestock. For each animal group except dairy cows, VS production was calculated using a national average VS production rate from the *Agricultural Waste Management Field Handbook* (USDA 1996b), which was then multiplied by the average weight of the animal and the State-specific animal population. For dairy cows, the national average VS constant was replaced with a mathematical relationship between milk production and VS, which was then multiplied by State-specific average annual milk production (USDA 2000j). The resulting VS for each animal group was then multiplied by the maximum methane producing capacity of the waste (B_o), and the State-specific methane conversion factors.

Nitrous oxide emissions were estimated by determining total Kjeldahl nitrogen (TKN)[2] production for all livestock wastes using livestock population data and nitrogen excretion rates. For each animal group, TKN production was calculated using a national average nitrogen excretion rate from the *Agricultural Waste Management Field Handbook* (USDA 1996b), which was then multiplied by the average weight of the animal and the State-specific animal population. State-specific weighted N_2O emission factors specific to the type of manure management system were then applied to total nitrogen production to estimate N_2O emissions.

See Annex K for more detailed information on the methodology and data used to calculate methane and nitrous oxide emissions from manure management.

Data Sources

Animal population data for all livestock types, except horses and goats, were obtained from the U.S. Department of Agriculture's National Agricultural Statistics Service (USDA 1994a-b, 1995a-b, 1998a-b, 1999a-c, 2000a-

[2] Total Kjeldahl nitrogen is a measure of organically bound nitrogen and ammonia nitrogen.

g). Horse population data were obtained from the FAOSTAT database (FAO 2000). Goat population data were obtained from the Census of Agriculture (USDA 1999d). Information regarding poultry turnover (i.e., slaughter) rate was obtained from State Natural Resource Conservation Service (NRCS) personnel (Lange 2000). Dairy cow and swine population data by farm size for each State, used for the weighted MCF and emission factor calculations, were obtained from the *Census of Agriculture*, which is conducted every five years (USDA 1999e).

Manure management system usage data for dairy and swine operations were obtained from USDA's Centers for Epidemiology and Animal Health (USDA 1996b, 1998d, 2000h) for small operations and from preliminary estimates for EPA's Office of Water regulatory effort for large operations (ERG 2000). Data for poultry layers were obtained from a voluntary United Egg Producers' survey (UEP 1999), previous EPA estimates (EPA 1992), and USDA's Animal Plant Health Inspection Service (USDA 2000i). Data for beef feedlots were also obtained from EPA's Office of Water (ERG 2000). Manure management system usage data for other livestock were taken from previous EPA estimates (EPA 1992). Data regarding the use of daily spread and pasture, range, or paddock systems for dairy cattle were obtained from personal communications with personnel from several organizations, and data provided by those personnel (Poe et al. 1999). These organizations include State NRCS offices, State extension services, State universities, USDA National Agriculture Statistics Service (NASS), and other experts (Deal 2000, Johnson 2000, Miller 2000, Stettler 2000, Sweeten 2000, and Wright 2000). Additional information regarding the percent of beef steer and heifers on feedlots was obtained from contacts with the national USDA office (Milton 2000).

Volatile solids and nitrogen excretion rate data from the USDA Agricultural Waste Management Field Handbook (USDA 1996a) were used for all livestock except sheep, goats, and horses. Data from the American Society of Agricultural Engineers (ASAE 1999) were used for these animal types. In addition, annual NASS data for average milk production per cow per State (USDA 2000j) were used to calculate State-specific volatile solids production rates for dairy cows for each year. Nitrous oxide emission factors and MCFs for dry systems were taken from *Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories* (IPCC 2000). Methane conversion factors for liquid/slurry systems were calculated based on average ambient temperatures of the counties in which animal populations were located.

Uncertainty

The primary factors contributing to the uncertainty in emission estimates are a lack of information on the usage of various manure management systems in each regional location and the exact methane generating characteristics of each type of manure management system. Because of significant shifts toward larger swine and dairy farms, it is believed that increasing amounts of manure are being managed in liquid manure management systems. The existing estimates reflect these shifts in the weighted MCFs based on the 1992 and 1997 farm-size data. However, the assumption of a direct relationship between farm size and liquid system usage may not apply in all cases and may vary based on geographic location. In addition, the CH_4 generating characteristics of each manure management system type are based on relatively few laboratory and field measurements, and may not match the diversity of conditions under which manure is managed nationally.

IPCC (2000) published default CH_4 conversion factors of 0 to 100 percent for anaerobic lagoon systems, which reflects the wide range in performance that may be achieved with these systems. There exist relatively few data points on which to determine country-specific MCFs for these systems. In the United States, many livestock waste treatment systems classified as anaerobic lagoons are actually holding ponds that are substantially organically overloaded and therefore not producing methane at the same rate as a properly designed lagoon. In addition, these systems may not be well operated, contributing to higher loading rates when sludge is allowed to enter the treatment portion of the lagoon or the lagoon volume is pumped too low to allow treatment to occur. Rather than setting the MCF for all anaerobic lagoon systems in the United States based on data available from optimized lagoon systems, an MCF methodology was developed that more closely matches observed system performance and accounts for the affect of temperature on system performance.

However, there is uncertainty related to the new methodology. The MCF methodology used includes a factor to account for management and design practices that result in the loss of volatile solids from the management system. This factor is currently estimated based on data from anaerobic lagoons in temperate climates, and from only three systems. However, this methodology is intended to account for systems across a range of management practices. Future work in gathering measurement data from animal waste lagoon systems across the country will contribute to the verification and refinement of this methodology. It will also be evaluated whether lagoon temperatures differ substantially from ambient temperatures and whether a lower bound estimate of temperature should be established for use with this methodology.

The IPCC provides a suggested MCF for poultry waste management operations of 1.5 percent. Additional study is needed in this area to determine if poultry high rise houses promote sufficient aerobic conditions to warrant a lower MCF.

The default N_2O emission factors published in *Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories* (IPCC 2000) were derived using limited information. The IPCC factors are global averages; U.S.-specific emission factors may be significantly different. Manure and urine in anaerobic lagoons and liquid/slurry management systems produce methane at different rates, and would in all likelihood produce nitrous oxide at different rates, although a single N_2O emission factors was used for both system types. In addition, there are little data available to determine the extent to which nitrification-denitrification occurs in animal waste management systems. Ammonia concentrations that are present in poultry and swine systems suggest that the N_2O emission estimates may be high. Current research to measure N_2O from liquid manure systems also suggests that these emissions may be overstated. At this time, there are insufficient data available to develop U.S.-specific N_2O emission factors; however, this is an area of on-going research, and warrants further study as more data become available.

Although an effort was made to introduce the variability in volatile solids production due to differences in diet for dairy cows, additional work is needed to establish the relationship between milk production and volatile solids production. In addition, the corresponding dairy methane emissions may be underestimated because milk production was unable to be correlated to specific manure management systems in each State. A methodology to assess variability in swine volatile solids production would be useful in future inventory estimates.

Uncertainty also exists with the maximum CH_4 producing potential of volatile solids excreted by different animal groups (i.e., B_o). The B_o values used in the CH_4 calculations are published values for U.S. animal waste. However, there are several studies that provide a range of B_o values for certain animals, including dairy and swine. Separate B_o values for dairy cows and dairy heifers were chosen to better represent the feeding regimens of these animal groups. For example, dairy heifers do not receive an abundance of high energy feed and consequently, dairy heifer manure will not produce as much CH_4 as manure from a milking cow. However, the data available for B_o values are sparse, and do not necessarily reflect the rapid changes that have occurred in this industry with respect to feed regimens. Further investigation to these waste characteristics is an area for further improvement.

Rice Cultivation

Most of the world's rice, and all rice in the United States, is grown on flooded fields. When fields are flooded, aerobic decomposition of organic material gradually depletes the oxygen present in the soil and floodwater, causing anaerobic conditions in the soil to develop. Once the environment becomes anaerobic, methane is produced through anaerobic decomposition of soil organic matter by methanogenic bacteria. As much as 60 to 90 percent of the methane produced is oxidized by aerobic methanotrophic bacteria in the soil (Holzapfel-Pschorn et al. 1985, Sass et al. 1990). Some of the methane is also leached away as dissolved methane in floodwater that percolates from the field. The remaining un-oxidized methane is transported from the submerged soil to the atmosphere primarily by diffusive transport through the rice plants. Some methane also escapes from the soil via diffusion and bubbling through floodwaters.

The water management system under which rice is grown is one of the most important factors affecting methane emissions. Upland rice fields are not flooded, and therefore are not believed to produce methane. In deepwater rice fields (i.e., fields with flooding depths greater than one meter), the lower stems and roots of the rice plants are dead so the primary methane transport pathway to the atmosphere is blocked. The quantities of methane released from deepwater fields, therefore, are believed to be significantly less than the quantities released from areas with more shallow flooding depths. Some flooded fields are drained periodically during the growing season, either intentionally or accidentally. If water is drained and soils are allowed to dry sufficiently, methane emissions decrease or stop entirely. This is due to soil aeration, which not only causes existing soil methane to oxidize but also inhibits further methane production in soils. All rice in the United States is grown under continuously flooded conditions; none is grown under deepwater conditions. Mid-season drainage does not occur except by accident (e.g., due to levee breach).

Other factors that influence methane emissions from flooded rice fields include fertilization practices (especially the use of organic fertilizers), soil temperature, soil type, rice variety, and cultivation practices (e.g., tillage, and seeding and weeding practices). The factors that determine the amount of organic material that is available to decompose (i.e., organic fertilizer use, soil type, rice variety,[3] and cultivation practices) are the most important variables influencing methane emissions over an entire growing season because the total amount of methane released depends primarily on the amount of organic substrate available. Soil temperature is known to be an important factor regulating the activity of methanogenic bacteria, and therefore the rate of methane production. However, although temperature controls the amount of time it takes to convert a given amount of organic material to methane, that time is short relative to a growing season, so the dependence of emissions over an entire growing season on soil temperature is weak. The application of synthetic fertilizers has also been found to influence methane emissions; in particular, both nitrate and sulfate fertilizers (e.g., ammonium nitrate, and ammonium sulfate) appear to inhibit methane formation.

Rice is cultivated in seven States: Arkansas, California, Florida, Louisiana, Mississippi, Missouri, and Texas. Soil types, soil temperatures, rice varieties, and cultivation practices for rice vary from State to State, and even from farm to farm. However, most rice farmers utilize organic fertilizers in the form of rice residue from the previous crop, which is left standing, disked, or rolled into the fields. Most farmers also apply synthetic fertilizer to their fields, usually urea. Nitrate and sulfate fertilizers are not commonly used in rice cultivation in the United States. In addition, the climatic conditions of Arkansas, southwest Louisiana, Texas, and Florida allow for a second, or ratoon, rice crop. This second rice crop is produced from regrowth of the stubble after the first crop has been harvested. Because the first crop's stubble is left behind in ratooned fields, the amount of organic material that is available for decomposition is considerably higher than with the first (i.e., primary) crop. Methane emissions from ratoon crops have been found to be considerably higher than those from the primary crop.

Rice cultivation is a small source of methane in the United States (Table 5-8 and Table 5-9). In 1999, methane emissions from rice cultivation were 10.7 Tg CO_2 Eq. (509 Gg)—only about 2 percent of total U.S. methane emissions. Although annual emissions fluctuated up and down between the years 1990 and 1999, there was a general increase over the nine year period due to an increase in harvested area. Between 1990 and 1999, total emissions increased by 23 percent.

The factors that affect the rice acreage harvested in any year vary from State to State. In Florida, the State having the smallest harvested rice area, rice acreage is largely a function of sugarcane acreage. Sugarcane fields are flooded each year after harvest to control pests, and on this flooded land a rice crop is grown along with a ratoon crop of sugarcane (Schueneman 1997). In Mis-

[3] The roots of rice plants shed organic material, which is referred to as "root exudate." The amount of root exudate produced by a rice plant over a growing season varies among rice varieties.

Table 5-8: CH_4 Emissions from Rice Cultivation (Tg CO_2 Eq.)

State	1990	1995	1996	1997	1998	1999
Arkansas	2.5	2.8	2.5	2.9	3.2	3.5
California	1.5	1.8	1.9	2.0	1.8	2.1
Florida	0.1	0.1	0.1	0.1	0.1	0.1
Louisiana	2.7	2.8	2.6	2.9	3.0	3.0
Mississippi	0.5	0.6	0.5	0.5	0.6	0.7
Missouri	0.2	0.3	0.3	0.3	0.4	0.5
Texas	1.2	1.0	1.0	0.8	0.9	0.8
Total	**8.7**	**9.5**	**8.8**	**9.6**	**10.1**	**10.7**

Note: Totals may not sum due to independent rounding.

Table 5-9: CH_4 Emissions from Rice Cultivation (Gg)

State	1990	1995	1996	1997	1998	1999
Arkansas	121	135	118	140	154	16
California	72	85	91	94	87	98
Florida	3	5	5	4	4	4
Louisiana	127	133	125	136	145	144
Mississippi	26	30	22	25	28	34
Missouri	10	14	12	15	18	23
Texas	55	50	47	40	44	40
Total	**414**	**452**	**419**	**455**	**481**	**509**

Note: Totals may not sum due to independent rounding.

souri, rice acreage is affected by weather (e.g., rain during the planting season may prevent the planting of rice), the price differential between soybeans and rice (i.e., if soybean prices are higher, then soybeans may be planted on some of the land which would otherwise have been planted in rice), and government support programs (Stevens 1997). The price differential between soybeans and rice also affects rice acreage in Mississippi. Rice in Mississippi is usually rotated with soybeans, but if soybean prices increase relative to rice prices, then some of the acreage that would have been planted in rice, is instead planted in soybeans (Street 1997). In Texas, rice production, and therefore harvested area, are affected by both government programs and the cost of production (Klosterboer 1997). California rice area is influenced by water availability as well as government programs and commodity prices. In Louisiana, rice area is influenced by government programs, weather conditions (e.g., rainfall during the planting season), as well as the price differential between rice and corn and other crops (Saichuk 1997). Arkansas rice area has been influenced in the past by government programs. However, due to the phase-out of these programs nationally, which began in 1996, spring commodity prices have had a greater effect on the amount of land planted in rice in recent years (Mayhew 1997).

Methodology

The *Revised 1996 IPCC Guidelines* (IPCC/UNEP/OECD/IEA 1997) recommend applying a seasonal emission factor to the annual harvested rice area to estimate annual CH_4 emissions. This methodology assumes that a seasonal emission factor is available for all growing conditions. Because season lengths are quite variable both within and among States in the United States, and because flux measurements have not been taken under all growing conditions in the United States, an earlier IPCC methodology (IPCC/UNEP/OECD/IEA 1995) has been applied here, using season lengths that vary slightly from the recommended approach. The 1995 *IPCC Guidelines* recommend multiplying a daily average emission factor by growing season length and annual harvested area. The *IPCC Guidelines* suggest that the "growing" season be used to calculate emissions based on the assumption that emission factors are derived from measurements over the whole growing season rather than just the flooding season. Applying this assumption to the United States, however, would result in an overestimate of emissions because the emission factors developed for the United States are based on measurements over the flooding, rather than the growing, season. Therefore, the method used here is based on the number of days of flooding during the growing season and a daily average emission factor, which is multiplied by the harvested area. Agricultural extension agents in each of the seven States in the United States that produce rice were contacted to determine water management practices and flooding season lengths in each State. Although all contacts reported that rice growing areas were continuously flooded, flooding season lengths varied considerably among States; therefore, emissions were calculated separately for each State.

Emissions from ratooned and primary areas are estimated separately. Information on ratoon flooding season lengths was collected from agricultural extension agents in the States that practice ratooning, and emission factors for both the primary season and the ratoon season were derived from published results of field experiments in the United States.

Data Sources

The harvested rice areas for the primary and ratoon crops in each State are presented in Table 5-10. Data for 1990 through 1999 for all States except Florida were taken from *U.S. Department of Agriculture's National Agriculture Statistics Data—Published Estimates Database* (USDA 2000). Harvested rice areas in Florida from 1990 to 1999 were obtained from Tom Schueneman (1999b, 1999c, 2000), a Florida Agricultural Extension Agent. Acreages for the ratoon crops were derived from conversations with the agricultural extension agents in each State. In Arkansas, ratooning occurred only in 1998 and 1999, when the ratooned area was less than 1 percent of the primary area (Slaton 1999a, 2000). In Florida, the ratooned area was 50 percent of the primary area from 1990 to 1998 (Schueneman 1999a) and about 65 percent of the primary area in 1999 (Schueneman 2000). In the other two States in which ratooning is practiced (i.e., Louisiana and Texas), the percentage of the primary area that was ratooned was constant over the entire 1990 to 1999 period. In Louisiana it was 30 percent (Linscombe 1999a, Bollich 2000), and in Texas it was 40 percent (Klosterboer 1999a, 2000).

Information about flooding season lengths was obtained from agricultural extension agents in each State (Beck 1999, Guethle 1999, Klosterboer 1999b, Linscombe 1999b, Scardaci 1999a and 1999b, Schueneman 1999b, Slaton 1999b, Street 1999a and 1999b). These data were assumed to apply to 1990 through 1999, and are presented in Table 5-11.

To determine what daily methane emission factors should be used for the primary and ratoon crops, methane flux information from rice field measurements in the United States was collected. Experiments which involved the application of nitrate or sulfate fertilizers, or other substances believed to suppress methane formation, as well as experiments in which measurements were not made over an entire flooding season or in which floodwaters were drained mid-season, were excluded from the analysis. This process left ten field experiments from California (Cicerone et al. 1992), Texas (Sass et al. 1990, 1991a, 1991b, 1992), and Louisiana (Lindau et al. 1991, Lindau and Bollich 1993, Lindau et al. 1993, Lindau et al. 1995, Lindau et al. 1998).[4] These experimental results were then sorted by season and type of fertilizer amendment (i.e., no fertilizer added, organic fertilizer added, and synthetic and organic fertilizer added). The results for the primary crop showed no consistent correlation between emission rate and type or magnitude of fertilizer application. Although individual experiments have shown a significant increase in emissions when organic fertilizers are added, when the results were combined, emissions from fields that received organic fertilizers were not found to be, on average, higher

[4] In some of these remaining experiments, measurements from individual plots were excluded from the analysis because of the reasons just mentioned. In addition, one measurement from the ratooned fields (i.e., the flux of 2.041 g/m^2/day in Lindau and Bollich 1993) was excluded since this emission rate is unusually high compared to other flux measurements in the United States, as well as in Europe and Asia (IPCC/UNEP/OECD/IEA 1997).

Table 5-10: Rice Areas Harvested (Hectares)

State/Crop	1990		1995	1996	1997	1998	1999
Arkansas							
Primary	485,633		542,291	473,493	562,525	617,159	665,722
Ratoon*	NO		NO	NO	NO	202	202
California	159,854		188,183	202,347	208,822	193,444	216,512
Florida							
Primary	4,978		9,713	8,903	7,689	8,094	7,229
Ratoon	2,489		4,856	4,452	3,845	4,047	4,673
Louisiana							
Primary	220,558		230,676	215,702	235,937	250,911	249,292
Ratoon	66,168		69,203	64,711	70,781	75,273	74,788
Mississippi	101,174		116,552	84,176	96,317	108,458	130,716
Missouri	32,376		45,326	38,446	47,349	57,871	74,464
Texas							
Primary	142,857		128,693	120,599	104,816	114,529	104,816
Ratoon	57,143		51,477	48,240	41,926	45,811	41,926
Total	**1,273,229**		**1,386,969**	**1,261,068**	**1,380,008**	**1,475,799**	**1,570,340**

Note: Totals may not sum due to independent rounding.
* Arkansas ratooning occurred only in 1998 and 1999.
NO (Not Occurring)

Table 5-11: Rice Flooding Season Lengths (Days)

State/Crop	Low	High
Arkansas		
Primary	60	80
Ratoon	30	40
California	100	145
Florida		
Primary	90	110
Ratoon	40	60
Louisiana		
Primary	90	120
Ratoon	70	75
Mississippi	68	82
Missouri	80	100
Texas		
Primary	60	80
Ratoon	40	60

that those from fields that receive synthetic fertilizer only. In addition, there appeared to be no correlation between fertilizer application rate and emission rate, either for synthetic or organic fertilizers. These somewhat surprising results are probably due to other variables that have not been taken into account, such as timing and mode of fertilizer application, soil type, cultivar type, and other cultivation practices. There were limited results from ratooned fields. Of those that received synthetic fertilizers, there was no consistent correlation between emission rate and amount of fertilizer applied. All the ratooned fields that received synthetic fertilizer had emission rates that were higher than the one ratoon experiment in which no synthetic fertilizer was applied. Given these results, the lowest and highest emission rates measured in primary fields that received synthetic fertilizer only—which bounded the results from fields that received both synthetic and organic fertilizers—were used as the emission factor range for the primary crop, and the lowest and highest emission rates measured in all the ratooned fields were used as the emission factor range for the ratoon crop. These ranges are 0.020 to 0.609 g/m^2-day for the primary crop, and 0.301 to 0.933 g/m^2-day for the ratoon crop.

Uncertainty

The largest uncertainty in the calculation of CH_4 emissions from rice cultivation is associated with the emission factors. Daily average emissions, derived from field measurements in the United States, vary by more than one order of magnitude. This variability is due to differences in cultivation practices, particularly the type, amount, and mode of fertilizer application; differences in cultivar type; and differences in soil and climatic conditions. By separating primary from ratooned areas, this Inventory has accounted for some of this. A range for both the primary (0.315 g/m^{2}day ±93 percent) and ratoon crop (0.617

g/m^{2}day ±51 percent) has been used in these calculations to reflect the remaining uncertainty. Based on this range, total methane emissions from rice cultivation in 1999 were estimated to have been approximately 1.6 to 19.8 Tg CO_2 Eq. (76 to 943 Gg), or 10.7 Tg CO_2 Eq. ±85 percent.

Two other sources of uncertainty are the flooding season lengths and ratoon areas used for each State. Flooding seasons in each State may fluctuate from year to year, and thus a range has been used to reflect this uncertainty. Even within a State, flooding seasons can vary by county and cultivar type (Linscombe 1999a). Data on the areas ratooned each year are not compiled regularly, so expert judgement was used to estimate these areas.

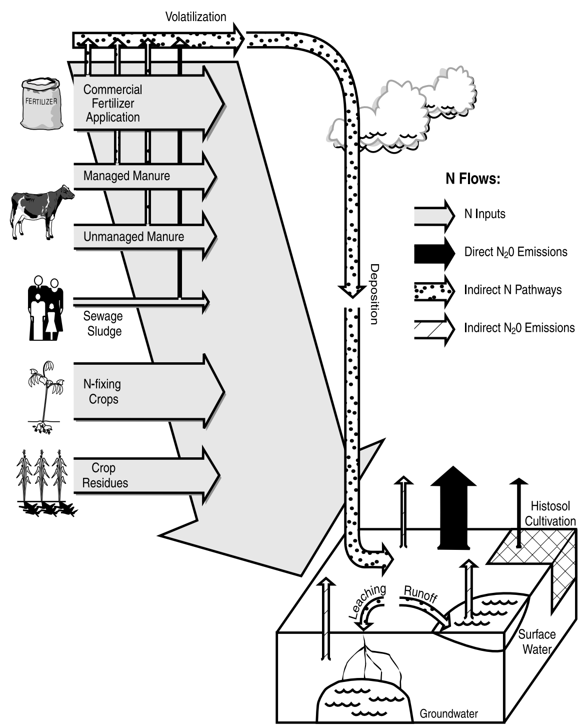
The last source of uncertainty is in the practice of flooding outside of the normal rice season. According to agriculture extension agents, all of the rice-growing States practice this on some part of their rice acreage, ranging from 5 to 33 percent of the rice acreage. Fields are flooded for a variety of reasons: to provide habitat for waterfowl, to provide ponds for crawfish production, and to aid in rice straw decomposition. To date, methane flux measurements have not been undertaken in these flooded areas, so this activity is not included in the emission estimates presented here.

Agricultural Soil Management

Nitrous oxide (N_2O) is produced naturally in soils through the microbial processes of nitrification and denitrification.[5] A number of agricultural activities add nitrogen to soils, thereby increasing the amount of nitrogen available for nitrification and denitrification, and ultimately the amount of N_2O emitted. These activities may add nitrogen to soils either directly or indirectly (Figure 5-2). Direct additions occur through various soil management practices and from the deposition of manure on soils by animals on pasture, range, and paddock (i.e., by animals whose manure is not managed). Soil management practices that add nitrogen to soils include fertilizer use, application of managed livestock manure, disposal of sewage sludge, production of nitrogen-fixing crops, application of crop residues, and cultivation of histosols (i.e., soils with a high organic matter content, otherwise known as organic soils).[6] Indirect additions of nitrogen to soils occur through two mechanisms: 1) volatilization and subsequent atmospheric

Figure 5-2

Sources of N_2O Emissions from Agricultural Soils



This graphic illustrates the sources and pathways of nitrogen that result in direct and indirect N_2O emissions from agricultural soils in the United States. Sources of nitrogen applied to, or deposited on, soils are represented with arrows on the left-hand side of the graphic. Emission pathways are also shown with arrows. On the lower right-hand side is a cut-away view of a representative section of managed soil; histosol cultivation is represented here.

[5] Nitrification and denitrification are two processes within the nitrogen cycle that are brought about by certain microorganisms in soils. Nitrification is the aerobic microbial oxidation of ammonium (NH_4) to nitrate (NO_3), and denitrification is the anaerobic microbial reduction of nitrate to dinitrogen gas (N_2). Nitrous oxide is a gaseous intermediate product in the reaction sequence of denitrification, which leaks from microbial cells into the soil and then into the atmosphere. Nitrous oxide is also produced during nitrification, although by a less well understood mechanism (Nevison 2000).

[6] Cultivation of histosols does not, *per se*, "add" nitrogen to soils. Instead, the process of cultivation enhances mineralization of old, nitrogen-rich organic matter that is present in histosols, thereby enhancing N_2O emissions from histosols.

deposition of applied nitrogen;[7] and 2) surface runoff and leaching of applied nitrogen into groundwater and surface water. Other agricultural soil management practices, such as irrigation, drainage, tillage practices, and fallowing of land, can affect fluxes of N_2O, as well as other greenhouse gases, to and from soils. However, because there are significant uncertainties associated with these other fluxes, they have not been estimated.

Agricultural soil management is the largest source of N_2O in the United States.[8] Estimated emissions from this source in 1999 are 298.3 Tg CO_2 Eq. (962 Gg), or approximately 69 percent of total U.S. N_2O emissions. Although annual agricultural soil management emissions fluctuated between 1990 and 1999, there was a general increase in emissions over the ten-year period (Table 5-12 and Table 5-13).[9] This general increase in emissions was due primarily to an increase in synthetic fertilizer use, manure production, and crop production over this period. The year-to-year fluctuations are largely a reflection of annual variations in synthetic fertilizer consumption and crop production. Over the ten-year period, total emissions of N_2O from agricultural soil management increased by approximately 11 percent. Estimated emissions, by subsource, are provided in Table 5-14, Table 5-15, and Table 5-16.

Methodology

The methodology used to estimate emissions from agricultural soil management is consistent with the *Revised 1996 IPCC Guidelines* (IPCC/UNEP/OECD/IEA 1997), as amended by the IPCC *Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories* (IPCC 2000). The *Revised 1996 IPCC Guidelines* divide this N_2O source category into three components: (1) direct emissions from managed soils due to applied nitrogen and cultivation of histosols; (2) direct emissions from soils due to the deposition of manure by livestock on pasture, range, and paddock; and (3) indirect emissions from soils induced by applied nitrogen.

Annex L provides more detailed information on the methodologies and data used to calculate N_2O emissions from each of these three components.

Direct N_2O Emissions from Managed Soils

Direct N_2O emissions from managed soils are composed of two parts, which are estimated separately and then summed. These two parts are 1) emissions due to nitrogen applications, and 2) emissions from histosol cultivation.

Estimates of direct N_2O emissions from nitrogen applications were based on the total amount of nitrogen that is applied to soils annually through the following practices: (a) the application of synthetic and organic commercial fertilizers, (b) the application of livestock manure through both daily spread operations and through the eventual application of manure that had been stored in manure management systems, (c) the application of sewage sludge, (d) the production of nitrogen-fixing crops, and (e) the application of crop residues. For each of these practices, the annual amounts of nitrogen applied were estimated as follows:

a) Synthetic and organic commercial fertilizer nitrogen applications were derived from annual fertilizer consumption data and the nitrogen content of the fertilizers.

b) Livestock manure nitrogen applications were based on the assumption that all livestock manure is applied to soils except for two components: 1) a small portion of poultry manure that is used as a livestock feed supplement; and 2) the manure from pasture, range, and paddock livestock. The manure nitrogen data were derived from animal population and weight statistics, information on manure management system usage, annual nitrogen excretion rates for each animal type, and information on the fraction of poultry litter that is used as a livestock feed supplement.

[7] These processes entail volatilization of applied nitrogen as ammonia (NH_3) and oxides of nitrogen (NO_x), transformations of these gases within the atmosphere (or upon deposition), and deposition of the nitrogen primarily in the form of particulate ammonium (NH_4), nitric acid (HNO_3), and oxides of nitrogen.

[8] Note that the emission estimates for this source category include applications of nitrogen to *all* soils, but the term "Agricultural Soil Management" is kept for consistency with the reporting structure of the *Revised 1996 IPCC Guidelines* (IPCC/UNEP/OECD/IEA 1997).

[9] Emission estimates for all years are presented in Annex L.

Table 5-12: N_2O Emissions from Agricultural Soil Management (Tg CO_2 Eq.)

Activity	1990	1995	1996	1997	1998	1999
Direct	**195.1**	**206.4**	**213.9**	**219.4**	**220.1**	**218.0**
Managed Soils	154.4	162.4	170.0	176.8	178.4	176.6
Pasture, Range, & Paddock Livestock	40.7	44.0	43.9	42.6	41.8	41.4
Indirect	73.9	79.0	80.7	80.4	80.2	80.3
Total	**269.0**	**285.4**	**294.6**	**299.8**	**300.3**	**298.3**

Note: Totals may not sum due to independent rounding.

Table 5-13: N_2O Emissions from Agricultural Soil Management (Gg)

Activity	1990	1995	1996	1997	1998	1999
Direct	**629**	**666**	**690**	**708**	**710**	**703**
Managed Soils	498	524	549	570	575	570
Pasture, Range, & Paddock Livestock	131	142	142	138	135	133
Indirect	**238**	**255**	**260**	**259**	**259**	**259**
Total	**868**	**921**	**950**	**967**	**969**	**962**

Note: Totals may not sum due to independent rounding.

Table 5-14: Direct N_2O Emissions from Managed Soils (Tg CO_2 Eq.)

Activity	1990	1995	1996	1997	1998	1999
Commercial Fertilizers*	55.4	59.2	61.2	61.3	61.4	61.8
Livestock Manure	12.7	13.2	13.4	13.7	13.8	13.8
Sewage Sludge	0.5	0.7	0.7	0.7	0.7	0.7
N Fixation	58.6	62.0	64.0	68.2	69.3	68.2
Crop Residue	23.3	23.4	26.9	29.1	29.3	28.3
Histosol Cultivation	3.9	3.9	3.9	3.9	3.9	3.9
Total	**154.4**	**162.5**	**170.1**	**176.9**	**178.5**	**176.7**

Note: Totals may not sum due to independent rounding.
* Excludes sewage sludge and livestock manure used as commercial fertilizers.

Table 5-15: Direct N_2O Emissions from Pasture, Range, and Paddock Livestock Manure (Tg CO_2 Eq.)

Animal Type	1990	1995	1996	1997	1998	1999
Beef Cattle	35.2	38.9	39.0	37.8	37.0	36.7
Dairy Cows	1.7	1.5	1.4	1.3	1.3	1.2
Swine	+	+	+	+	+	+
Sheep	+	+	+	+	+	+
Goats	+	+	+	+	+	+
Poultry	+	+	+	+	+	+
Horses	2.5	2.7	2.7	2.7	2.7	2.7
Total	**40.7**	**44.0**	**43.9**	**42.6**	**41.8**	**41.4**

Note: Totals may not sum due to independent rounding.
+ Less than 0.5 Tg CO_2 Eq.

Table 5-16: Indirect N_2O Emissions (Tg CO_2 Eq.)

Activity	1990		1995	1996	1997	1998	1999
Volatilization & Atm. Deposition	**11.7**		**12.5**	**12.7**	**12.6**	**12.6**	**12.6**
Commercial Fertilizers*	4.9		5.3	5.4	5.5	5.5	5.5
Livestock Manure	6.6		7.1	7.1	7.0	7.0	6.9
Sewage Sludge	+		+	+	+	+	+
Surface Leaching & Runoff	**62.2**		**66.5**	**68.0**	**67.8**	**67.6**	**67.7**
Commercial Fertilizers*	36.9		39.5	40.8	40.9	40.9	41.2
Livestock Manure	24.9		26.5	26.6	26.4	26.1	26.0
Sewage Sludge	+		0.5	0.5	0.6	0.6	0.6
Total	**73.9**		**79.0**	**80.7**	**80.4**	**80.2**	**80.3**

Note: Totals may not sum due to independent rounding.
* Excludes sewage sludge and livestock manure used as commercial fertilizers.
\+ Less than 0.5 Tg CO_2 Eq.

c) Sewage sludge nitrogen applications were derived from estimates of annual U.S. sludge production, the nitrogen content of the sludge, and periodic surveys of sludge disposal methods.

d) The amounts of nitrogen made available to soils through the cultivation of nitrogen-fixing crops were based on estimates of the amount of nitrogen in aboveground plant biomass, which were derived from annual crop production statistics, mass ratios of aboveground residue to crop product, dry matter fractions, and nitrogen contents of the plant biomass.

e) Crop residue nitrogen applications were derived from information about which residues are typically left on the field, the fractions of residues left on the field, annual crop production statistics, mass ratios of aboveground residue to crop product, and dry matter fractions and nitrogen contents of the residues.

After the annual amounts of nitrogen applied were estimated for each practice, each amount of nitrogen was reduced by the fraction that is assumed to volatilize according to the *Revised 1996 IPCC Guidelines* and the IPCC *Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories*. The net amounts left on the soil from each practice were then summed to yield total unvolatilized applied nitrogen, which was multiplied by the IPCC default emission factor for nitrogen applications.

Estimates of annual N_2O emissions from histosol cultivation were based on estimates of the total U.S. acreage of histosols cultivated annually. To estimate annual emissions, these areas were multiplied by the IPCC default emission factor for temperate histosols.[10]

Total annual emissions from nitrogen applications, and annual emissions from histosol cultivation, were then summed to estimate total direct emissions from managed soils.

Direct N_2O Emissions from Pasture, Range, and Paddock Livestock Manure

Estimates of N_2O emissions from this component are based on amounts of nitrogen in the manure that is deposited annually on soils by livestock in pasture, range, and paddock. Estimates of annual manure nitrogen from these livestock were derived from animal population and weight statistics; information on the fraction of the total population of each animal type that is on pasture, range, or paddock; and annual nitrogen excretion rates for each animal type. The annual amounts of manure nitrogen from each animal type were summed over all animal types to yield total pasture, range, and paddock manure nitrogen, which was then multiplied by the IPCC default emission factor for pasture, range, and paddock nitrogen to estimate N_2O emissions.

[10] Note that the IPCC default emission factors for histosols have been revised in the IPCC *Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories* (IPCC 2000). The revised default emission factor for temperate histosols (IPCC 2000) was used in these calculations.

Indirect N_2O Emissions from Soils

Indirect emissions of N_2O are composed of two parts, which are estimated separately and then summed. These two parts are 1) emissions resulting from volatilization and subsequent deposition of the nitrogen in applied fertilizers, applied sewage sludge, and all livestock manure, and 2) leaching and runoff of nitrogen in applied fertilizers, applied sewage sludge, and all livestock manure.[11] The activity data (i.e., nitrogen in applied fertilizers, applied sewage sludge, and all livestock manure) are the same for both parts, and were estimated in the same way as for the direct emission estimates.

To estimate the annual amount of applied nitrogen that volatilizes, the annual amounts of applied synthetic fertilizer nitrogen, applied sewage sludge nitrogen, and all livestock manure nitrogen, were each multiplied by the appropriate IPCC default volatilization fraction. The three amounts of volatilized nitrogen were then summed, and the sum was multiplied by the IPCC default emission factor for volatilized/deposited nitrogen.

To estimate the annual amount of nitrogen that leaches or runs off, the annual amounts of applied synthetic fertilizer nitrogen, applied sewage sludge nitrogen, and all livestock manure nitrogen were each multiplied by the IPCC default leached/runoff fraction. The three amounts of leached/runoff nitrogen were then summed, and the sum was multiplied by the IPCC default emission factor for leached/runoff nitrogen.

Total annual indirect emissions from volatilization, and annual indirect emissions from leaching and runoff, were then summed to estimate total indirect emissions of N_2O from managed soils.

Data Sources

The activity data used in these calculations were obtained from numerous sources. Annual synthetic and organic fertilizer consumption data for the United States were obtained from annual publications on commercial fertilizer statistics (TVA 1991, 1992a, 1993, 1994; AAPFCO 1995, 1996, 1997, 1998, 1999). Fertilizer nitrogen contents were taken from these same publications or Terry (1997). Livestock population data were obtained from USDA publications (USDA 1994b,c; 1995a,b; 1998a,c; 1999a-e; 2000a-g), the FAOSTAT database (FAO 2000), and Lange (2000). Manure management information was obtained from Poe et al. (1999), Safley et al. (1992), and personal communications with agricultural experts (Anderson 2000, Deal 2000, Johnson 2000, Miller 2000, Milton 2000, Stettler 2000, Sweeten 2000, Wright 2000). Livestock weight data were obtained from Safley (2000), USDA (1996, 1998d), and ASAE (1999); daily rates of nitrogen excretion from ASAE (1999) and USDA (1996); and information about the fraction of poultry litter used as a feed supplement from Carpenter (1992). Data collected by the U.S. EPA were used to derive annual estimates of land application of sewage sludge (EPA 1993, Bastian 1999). The nitrogen content of sewage sludge was taken from National Research Council (1996). Annual production statistics for nitrogen-fixing crops were obtained from USDA reports (USDA 1994a, 1997, 1998b, 1999f, 2000i), a book on forage crops (Taylor and Smith 1995, Pederson 1995, Beuselinck and Grant 1995, Hoveland and Evers 1995), and personal communications with forage experts (Cropper 2000, Gerrish 2000, Hoveland 2000, Evers 2000, and Pederson 2000). Mass ratios of aboveground residue to crop product, dry matter fractions, and nitrogen contents for nitrogen-fixing crops were obtained from Strehler and Stützle (1987), Barnard and Kristoferson (1985), Karkosh (2000), Ketzis (1999), and IPCC/UNEP/OECD/IEA (1997). Annual production statistics for crops whose residues are left on the field were obtained from USDA reports (USDA 1994a, 1997, 1998b, 1999f). Aboveground residue to crop mass ratios, residue dry matter fractions, and residue nitrogen contents were obtained from Strehler and Stützle (1987), Turn et al. (1997), and Ketzis (1999). Estimates of the fractions of residues left on the field were based on information provided by Karkosh (2000), and on information about rice residue burning (see the Agricultural Residue Burning section). The annual areas of cultivated histosols were estimated from 1982, 1992, and 1997 statistics in USDA's 1992 and 1997 National Resources Inventories (USDA 1994d and 2000h, as cited in Paustian 1999 and Sperow 2000, respectively).

[11] Total livestock manure nitrogen is used in the calculation of indirect N_2O emissions because all manure nitrogen, regardless of how the manure is managed or used, is assumed to be subject to volatilization and leaching and runoff.

All emission factors, volatilization fractions, and the leaching/runoff fraction were taken from the *Revised 1996 IPCC Guidelines* (IPCC/UNEP/OECD/IEA 1997), as amended by the IPCC *Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories* (IPCC 2000).

Uncertainty

The amount of N_2O emitted from managed soils depends not only on N inputs, but also on a large number of variables, including organic carbon availability, O_2 partial pressure, soil moisture content, pH, and soil temperature. However, the effect of the combined interaction of these other variables on N_2O flux is complex and highly uncertain. Therefore, the IPCC default methodology, which is used here, is based only on N inputs and does not utilize these other variables. As noted in the *Revised 1996 IPCC Guidelines* (IPCC/UNEP/OECD/IEA 1997), this is a generalized approach that treats all soils as being under the same conditions. The estimated ranges around the IPCC default emission factors provide an indication of the uncertainty in the emission estimates due to this simplified methodology. Most of the emission factor ranges are about an order of magnitude, or larger. Developing an emission estimation methodology that explicitly utilizes these other variables will require more scientific research, and will likely involve the use of process models.

Uncertainties also exist in the activity data used to derive emission estimates. In particular, the fertilizer statistics include only those organic fertilizers that enter the commercial market, so non-commercial fertilizers (other than the estimated manure and crop residues) have not been captured. Also, the nitrogen content of organic fertilizers varies by type, as well as within individual types; however, average values were used to estimate total organic fertilizer nitrogen consumed. The livestock excretion values, while based on detailed population and weight statistics, were derived using simplifying assumptions concerning the types of management systems employed. Statistics on sewage sludge applied to soils were not available on an annual basis; annual production and application estimates were based on two data points that were calculated from surveys that yielded uncertainty levels as high as 14 percent (Bastian 1999). The production statistics for the nitrogen-fixing crops that are forage legumes are highly uncertain because statistics are not compiled for these crops except for alfalfa, and the alfalfa statistics include alfalfa mixtures. Conversion factors for the nitrogen-fixing crops were based on a limited number of studies, and may not be representative of all conditions in the United States. Data on crop residues left on the field are not available, so expert judgement was used to estimate the amount of residues applied to soils. And finally, the estimates of cultivated histosol areas are uncertain because they are from a natural resource inventory that was not explicitly designed as a soil survey. However, these areas are consistent with those used in the organic soils component of the Land-Use Change and Forestry Chapter. Also, all histosols were assigned to the temperate climate regime; however, some of these areas are in subtropical areas, and therefore may be experiencing somewhat higher emission rates.[12]

Agricultural Residue Burning

Large quantities of agricultural crop residues are produced by farming activities. There are a variety of ways to dispose of these residues. For example, agricultural residues can be left on or plowed back into the field, composted and then applied to soils, landfilled, or burned in the field. Alternatively, they can be collected and used as a fuel or sold in supplemental feed markets. Field burning of crop residues is not considered a net source of carbon dioxide (CO_2) because the carbon released to the atmosphere as CO_2 during burning is assumed to be reabsorbed during the next growing season. Crop residue burning is, however, a net source of methane (CH_4), nitrous oxide (N_2O), carbon monoxide (CO), and nitrogen oxides (NO_x), which are released during combustion.

Field burning is not a common method of agricultural residue disposal in the United States; therefore, emissions from this source are minor. The primary crop types whose residues are typically burned in the United States are wheat, rice, sugarcane, corn, barley, soybeans,

[12] As discussed in Annex L, these issues regarding histosols will be researched in future U.S. Inventories.

and peanuts, and of these residues, less than 5 percent is burned each year, except for rice.[13] Annual emissions from this source over the period 1990 through 1999 averaged approximately 0.6 Tg CO_2 Eq. (28 Gg) of CH_4, 0.4 Tg CO_2 Eq. (1 Gg) of N_2O, 740 Gg of CO, and 33 Gg of NO_x (see Table 5-17 and Table 5-18).

Methodology

The methodology for estimating greenhouse gas emissions from field burning of agricultural residues is consistent with the *Revised 1996 IPCC Guidelines* (IPCC/UNEP/OECD/IEA 1997). In order to estimate the amounts of carbon and nitrogen released during burning, the following equations were used:

Carbon Released = (Annual Crop Production) ×(Residue/Crop Product Ratio)×(Fraction of Residues Burned *in situ*) ×(Dry Matter Content of the Residue)×(Burning Efficiency)×(Carbon Content of the Residue)× (Combustion Efficiency)[14]

Nitrogen Released = (Annual Crop Production)×(Residue/Crop Product Ratio)×(Fraction of Residues Burned *in situ*)×(Dry Matter Content of the Residue)×(Burning Efficiency)×(Nitrogen Content of the Residue)× (Combustion Efficiency)

Emissions of CH_4 and CO were calculated by multiplying the amount of carbon released by the appropriate IPCC default emission ratio (i.e., CH_4-C/C or CO-C/C). Similarly, N_2O and NO_x emissions were calculated by multiplying the amount of nitrogen released by the appropriate IPCC default emission ratio (i.e., N_2O-N/N or NO_x-N/N).

Data Sources

The crop residues that are burned in the United States were determined from various State level greenhouse gas emission inventories (ILENR 1993, Oregon Department of Energy 1995, Wisconsin Department of Natural Resources 1993) and publications on agricultural

Table 5-17: Emissions from Agricultural Residue Burning (Tg CO_2 Eq.)

Gas/Crop Type	1990		1995	1996	1997	1998	1999
CH_4	**0.5**		**0.5**	**0.6**	**0.6**	**0.6**	**0.6**
Wheat	0.1		0.1	0.1	0.1	0.1	0.1
Rice	0.1		0.1	0.1	+	+	+
Sugarcane	+		+	+	+	+	+
Corn	0.2		0.2	0.3	0.3	0.3	0.3
Barley	+		+	+	+	+	+
Soybeans	0.1		0.1	0.1	0.2	0.2	0.2
Peanuts	+		+	+	+	+	+
N_2O	**0.4**		**0.4**	**0.4**	**0.4**	**0.5**	**0.4**
Wheat	+		+	+	+	+	+
Rice	+		+	+	+	+	+
Sugarcane	+		+	+	+	+	+
Corn	0.1		0.1	0.1	0.1	0.1	0.1
Barley	+		+	+	+	+	+
Soybeans	0.2		0.2	0.2	0.3	0.3	0.3
Peanuts	+		+	+	+	+	+
Total	**0.9**		**0.9**	**1.0**	**1.0**	**1.1**	**1.0**

+ Does not exceed 0.05 Tg CO_2 Eq.
Note: Totals may not sum due to independent rounding.

[13] The fraction of rice straw burned each year is significantly higher than that for other crops (see "Data Sources" discussion below).

[14] Burning Efficiency is defined as the fraction of dry biomass exposed to burning that actually burns. Combustion Efficiency is defined as the fraction of carbon in the fire that is oxidized completely to CO_2. In the methodology recommended by the IPCC, the "burning efficiency" is assumed to be contained in the "fraction of residues burned" factor. However, the number used here to estimate the "fraction of residues burned" does not account for the fraction of exposed residue that does not burn. Therefore, a "burning efficiency factor" was added to the calculations.

Table 5-18: Emissions from Agricultural Residue Burning (Gg)*

Gas/Crop Type	1990		1995	1996	1997	1998	1999
CH_4	**25**		**24**	**28**	**29**	**30**	**28**
Wheat	5		4	4	5	5	4
Rice	2		2	3	2	2	2
Sugarcane	1		1	1	1	1	1
Corn	11		10	13	12	13	13
Barley	1		1	1	1	1	+
Soybeans	6		6	7	8	8	8
Peanuts	+		+	+	+	+	+
N_2O	**1**		**1**	**1**	**1**	**1**	**1**
Wheat	+		+	+	+	+	+
Rice	+		+	+	+	+	+
Sugarcane	+		+	+	+	+	+
Corn	+		+	+	+	+	+
Barley	+		+	+	+	+	+
Soybeans	1		1	1	1	1	1
Peanuts	+		+	+	+	+	+
CO	**668**		**641**	**735**	**750**	**776**	**740**
Wheat	137		109	114	124	128	115
Rice	65		65	73	55	53	49
Sugarcane	18		20	19	21	22	23
Corn	282		263	328	328	347	336
Barley	16		13	15	13	13	11
Soybeans	148		167	183	207	211	203
Peanuts	2		2	2	2	2	2
NO_x	**28**		**28**	**32**	**33**	**34**	**33**
Wheat	4		3	3	3	3	3
Rice	2		2	3	2	2	2
Sugarcane	+		+	+	+	+	+
Corn	7		6	8	8	8	8
Barley	1		+	+	+	+	+
Soybeans	14		16	17	20	20	19
Peanuts	+		+	+	+	+	+

* Full molecular weight basis.
+ Does not exceed 0.5 Gg
Note: Totals may not sum due to independent rounding.

burning in the United States (Jenkins et al. 1992, Turn et al. 1997, EPA 1992).

Crop production data were taken from the USDA's *Field Crops, Final Estimates 1987-1992, 1992-1997* (USDA 1994, 1998) and *Crop Production 1999 Summary* (USDA 2000). The production data for the crop types whose residues are burned are presented in Table 5-19.

The percentage of crop residue burned was assumed to be 3 percent for all crops in all years, except rice, based on State inventory data (ILENR 1993, Oregon Department of Energy 1995, Noller 1996, Wisconsin Department of Natural Resources 1993, and Cibrowski 1996). Estimates of the percentage of rice acreage on which residue burning took place were obtained on a State-by-State basis from agricultural extension agents in each of the seven rice-producing States (Bollich 2000; Guethle 1999, 2000; Fife 1999; California Air Resources Board 1999; Klosterboer 1999a, 1999b, 2000; Linscombe 1999a, 1999b; Najita 2000; Schueneman 1999a, 1999b; Slaton 1999a, 1999b, 2000; Street 1999a, 1999b, 2000) (see Table 5-20 and Table 5-21). The estimates provided for Arkansas and Florida remained constant over the entire 1990 through 1999 period, while the estimates for all other States varied over the time series. For California, it was assumed that the annual percents of rice acreage burned in Sacramento Valley are representative of burning in the entire State, because the Valley accounts for over 95 percent of the rice acreage in California (Fife 1999). The annual percents of rice acreage burned in Sacramento Valley were obtained from a report of the California Air Resources

Table 5-19: Agricultural Crop Production (Thousand Metric Tons of Product)

Crop	1990		1995	1996	1997	1998	1999
Wheat	74,292		59,404	61,980	67,534	69,327	62,662
Rice[a]	7,080		7,887	7,784	8,300	8,530	9,546
Sugarcane	25,525		27,922	26,729	28,766	30,896	32,406
Corn[b]	201,534		187,970	234,518	233,864	247,882	239,719
Barley	9,192		7,824	8,544	7,835	7,667	6,137
Soybeans	52,416		59,174	64,780	73,176	74,598	71,928
Peanuts	1,635		1,570	1,661	1,605	1,798	1,755

[a] Does not include rice production in Florida because rice residues are not burned in Florida (see Table 5-20).
[b] Corn for grain (i.e., excludes corn for silage).

Table 5-20: Percentage of Rice Area Burned by State

State	Percent Burned 1990-1998	Percent Burned 1999
Arkansas	10	10
California	variable[a]	23
Florida[b]	0	0
Louisiana	6	0
Mississippi	5	10
Missouri	3.5	5
Texas	1	2

[a] Values provided in Table 5-21.
[b] Burning of crop residues is illegal in Florida.

Table 5-21: Percentage of Rice Area Burned

Year	California	United States
1990	75	16
1995	59	15
1996	63	17
1997	34	12
1998	33	11
1999	23	9

Board (1999). These values declined over the 1990 through 1999 period because of a legislated reduction in rice straw burning (see Table 5-21). To derive the national percentage of rice acreage burned each year, the acreages burned in each State were summed and then divided by total U.S. rice harvested area (Table 5-21).

All residue/crop product mass ratios except sugarcane were obtained from Strehler and Stützle (1987). The datum for sugarcane is from University of California (1977). Residue dry matter contents for all crops except soybeans and peanuts were obtained from Turn et al. (1997). Soybean dry matter content was obtained from Strehler and Stützle (1987). Peanut dry matter content was obtained through personal communications with Jen Ketzis (1999), who accessed Cornell University's Department of Animal Science's computer model, Cornell Net Carbohydrate and Protein System. The residue carbon contents and nitrogen contents for all crops except soybeans and peanuts are from Turn et al. (1997). The residue carbon content for soybeans and peanuts is the IPCC default (IPCC/UNEP/OECD/IEA 1997). The nitrogen content of soybeans is from Barnard and Kristoferson (1985). The nitrogen content of peanuts is from Ketzis (1999). These data are listed in Table 5-22. The burning efficiency was assumed to be 93 percent, and the combustion efficiency was assumed to be 88 percent, for all crop types (EPA 1994). Emission ratios for all gases (see Table 5-23) were taken from the *Revised 1996 IPCC Guidelines* (IPCC/UNEP/OECD/IEA 1997).

Uncertainty

The largest source of uncertainty in the calculation of non-CO_2 emissions from field burning of agricultural residues is in the estimates of the fraction of residue of each crop type burned each year. Data on the fraction burned, as well as the gross amount of residue burned each year, are not collected at either the national or State level. In addition, burning practices are highly variable among crops, as well as among States. The fractions of residue burned used in these calculations were based upon information collected by State agencies and in published literature. It is likely that these emission estimates will continue to change as more information becomes available in the future.

Table 5-22: Key Assumptions for Estimating Emissions from Agricultural Residue Burning*

Crop	Reisduce/Crop Ration	Fraction of Residue Burned	Dry Matter Fraction	Carbon Fraction	Nitrogen Fraction
Wheat	1.3	0.03	0.93	0.4428	0.0062
Rice	1.4	variable	0.91	0.3806	0.0072
Sugarcane	0.8	0.03	0.62	0.4235	0.0040
Corn	1.0	0.03	0.91	0.4478	0.0058
Barley	1.2	0.03	0.93	0.4485	0.0077
Soybeans	2.1	0.03	0.87	0.4500	0.0230
Peanuts	1.0	0.03	0.86	0.4500	0.0106

* The burning efficiency and combustion efficiency for all crops were assumed to be 0.93 and 0.88, respectively.

Other sources of uncertainty include the residue/crop product mass ratios, residue dry matter contents, burning and combustion efficiencies, and emission ratios. A residue/crop product ratio for a specific crop can vary among cultivars, and for all crops except sugarcane, generic residue/crop product ratios, rather than ratios specific to the United States, have been used. Residue dry matter contents, burning and combustion efficiencies, and emission ratios, all can vary due to weather and other combustion conditions, such as fuel geometry. Values for these variables were taken from literature on agricultural biomass burning.

Table 5-23: Greenhouse Gas Emission Ratios

Gas	Emission Ratio
CH_4[a]	0.004
CO[a]	0.060
N_2O[b]	0.007
NO_x[b]	0.121

[a] Mass of carbon compound released (units of C) relative to mass of total carbon released from burning (units of C).
[b] Mass of nitrogen compound released (units of N) relative to mass of total nitrogen released from burning (units of N).